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Review Paper

Australian acacias as invasive species: lessons to be learnt from regions with long planting histories§

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Problems associated with invasiveness of non-native tree species used in forestry are increasing rapidly worldwide and are most severe in areas with a long history of plantings. Lessons learnt in areas with long histories of plantings and invasions may be applicable to areas with shorter planting histories. Most research towards understanding such tree invasions has focused on *Pinus* species, though all groups of trees that have been widely used in forestry are invasive to some extent. This paper explores the experience of Australian *Acacia* species (wattles). Unlike some other groups of trees, no particular set of traits clearly separates highly invasive from less- or non-invasive wattles. All species that have been widely planted over a long period have become invasive; the extent of invasions is largely a function of human usage. These findings imply that propagule pressure in concert with residence times are the main drivers of invasiveness in wattles (many factors mediate these drivers, including fire, forest clearance and soil disturbance). The massive extent of recent plantings of Australian *Acacia* species in South-east Asia is therefore likely to result in large-scale invasions unless proactive management is implemented. The history of wattles in South Africa highlights the need for such proactive management. Wattles were of considerable net value to the South African economy immediately after introduction. However, the costs of wattle invasions increased over time to such an extent that (certainly over the last few decades) these costs exceed the benefits derived from the forestry industry. Wattles now dominate many natural ecosystems. We recommend several interventions to prevent a similar pattern in South-east Asia and to ensure the sustainability of plantation forestry based on wattles in the region. A spatially explicit assessment of invasion risk is required, and a monitoring system should be implemented. Cost–benefit analyses (that consider the full suite of perspectives relating to costs and benefits) need to be applied to determine the need for sustainable mitigation methods. Options for reducing potential invasiveness should be implemented; these include biological control targeting seed production (very good success has been achieved in South Africa) and the use of sterile cultivars.

Keywords: biological invasions, forestry, sustainable forestry, tree invasions, wattles

Introduction

Those who cannot remember the past are condemned to repeat it. George Santayana

Many trees used in different forms of forestry around the world cause problems as invasive species (Richardson 1998, 2011). This unwelcome by-product of forestry is emerging as a substantial environmental and economic cost to be factored in when considering the overall benefits of afforestation and when devising comprehensive strategies to enhance the sustainability of plantation forestry. Problems associated with biological invasions often occur long after the forestry industry has been established. The impacts are often felt in areas far removed from the plantations themselves, and the people most affected are often not the foresters or those who benefit from the plantations initially (van Wilgen and Richardson 2012).

Problems associated with invasions resulting from forestry plantations have increased rapidly in magnitude and complexity in recent decades with the significant expansion of forestry (Richardson et al. 1994; Simberloff et al. 2010; Richardson 2011; Felton et al. 2013). These problems have different facets and dimensions in different regions of the world due to, among other things, the range of species used, the types of forestry, socio-economic contexts, the residence time of plantings, the prevailing perceptions of conservation ethics and values, and the effectiveness of management interventions (Richardson 2011). Despite such differences, evidence is emerging that experience with invasions of forestry trees in regions with a longer history of forestry can be transferred to regions with shorter histories of plantings and that such insights can be used to plan interventions in advance and to

§This article is based on a paper presented at the 'Sustaining the Future of Acacia Plantation Forestry' IUFRO WP 2.08.07 conference, March 2014, Hue, Vietnam

Box 1: Lessons learnt from pine invasions in Australia, New Zealand and South Africa for areas with more recent pine plantation industries

Pines (genus *Pinus*) have emerged as an important model group for understanding the ecology of plant invasions for many of the same reasons as wattles are a model group, i.e. there are a large number of species in the genus, encompassing a wide range of ecological adaptations; many species are useful to humans and so have been widely planted; introductions and the fate of plantings are generally well documented; a few pine species are among the most widely used forestry species; and some pine invasions are also amongst the most iconic and damaging plant invasions. For these and other reasons, the global experiment of planting pines in novel environments has yielded many insights on the invasion ecology of the genus (Richardson 2006).

Twenty-four pine species are known to be invasive (Rejmánek and Richardson 2013). The occurrence of invasive species and the extent of invasion of these species have been strongly influenced by a range of factors that resulted in their introduction and shaped their dissemination in different regions (Richardson et al. 1994; Essl et al. 2010; McGregor et al. 2012; Procheş et al. 2012). In the Southern Hemisphere, regions with the oldest plantations and history of forestry have the most widespread pine invasions and the biggest problems with impacts associated with invasions. Australia, New Zealand and South Africa all have long histories of pine forestry and major problems with pine invasions (Richardson and Higgins 1998).

Insights from regions with long histories of pines as alien species, and of different forms of forestry and attempts to manage invasions, have much to offer other regions (such as Argentina, Brazil, Chile and Uruguay in South America) with shorter histories of pine forestry and where invasions are only beginning (Simberloff et al. 2010).

Among the key insights that are transferable from regions with longer histories of pine forestry to areas with shorter histories are the following:

- levels of inherent invasiveness are not equal for all species; several species become invasive wherever they are planted, given enough time (Richardson 2006)
- interactions between life-history traits and features of the receiving environment determine the dimensions and extent of invasions, and the types and magnitudes of impacts (Higgins and Richardson 1998)
- the extent of invasion is strongly influenced by factors that affect propagule pressure, such as the extent of planting and residence time (Richardson et al. 1994; McGregor et al. 2012).

However, despite the lessons learned, few foresters in South America have responded to the challenges of invasion. Control will likely end up being reactive.

reduce conflicts (Richardson et al. 2008; Simberloff et al. 2010). Most insights in this regard have come from pines (Box 1), but similar patterns are emerging for other tree species (e.g. the genus *Casuarina*; Potgieter et al. 2014). We address these issues here, focusing on the recent and considerable plantings of Australian acacias (wattles) in South-east Asia and the example posed by over a century of dealing with wattle invasions in South Africa.

Global biogeography and invasion ecology of wattles

Australian acacias (genus *Acacia*) are a useful model group for understanding invasions (Richardson et al. 2011; Dodet and Collet 2012; Kueffer et al. 2013). Some advantages of wattles in this regard are the very large number of introduced species in the genus (at least one-third of the c. 1 012 species have been moved by humans to areas outside their natural ranges), their wide utilisation for many purposes in many parts of their extra-Australian ranges, and the fact that introductions and the fate of plantings of wattles as exotics are generally well documented. It is therefore feasible to unravel the role of different factors in determining why some species escape cultivation and become invasive while others are less successful invasive species. We can also test the validity of particular paradigms associated with different introduction histories, e.g. multiple vs single introduction events, the role of various traits in determining invasiveness, and selective breeding and genetic enhancement (for further details see Richardson et al. 2011). In particular, long residence times and histories of management in some countries, but much shorter times in others, provides an opportunity to exchange lessons and build generalisations for best practice (Wilson et al. 2011).

Twenty-four *Acacia* species have been recorded as invasive in at least one location in the world, with some species being invasive in many more locations (Table 1). Of these, eight species are used for forestry; the rest were introduced and disseminated outside their native ranges for various other purposes, notably for sand stabilisation and for use as ornamentals. Importantly, there is a strong correlation between the extent of usage of wattles and the extent of invasions (Figure 1; Wilson et al. 2011). It is not clear whether this correlation is because those species that are most widely planted have traits that promote invasiveness (e.g. short generation times, high seed production and fast growth rates). Alternative explanations are that foresters actively select areas where invasions are most likely to occur, i.e. by selecting environmental conditions very similar to those in the native ranges of the species (Donaldson et al. 2014; Motloung et al. 2014), or that invasions are simply driven by their introduction effort (propagule pressure), i.e. more plantings over a longer period provide more chances to spread. High propagule pressure alleviates genetic constraints and Allee effects (density dependence/

Table 1: Distribution of Australian *Acacia* species as invasive alien species in 15 regions. #, Species used in forestry. Data from Rejmánek and Richardson (2013)

Acacia species	North America	Europe	Middle East	Asia	Indonesia	Pacific Islands	New Zealand	Australia	Indian Ocean islands	Africa (southern)	Africa (remainder)	Atlantic islands	South America	Caribbean islands	Central America
A. auriculiformis A.Cunn. ex Benth. #	*			*					*		*		*		
A. baileyana F.Muell.							*	*		*					
A. crassicarpa A.Cunn. ex Benth. #									*						
A. cyclops A.Cunn. ex G.Don	*	*	*					*		*					
A. dealbata Link #	*	*					*	*	*	*			*		
A. decurrens Willd. #							*	*		*	*				
A. elata A.Cunn. ex Benth.										*					
A. holosericea A.Cunn. ex G.Don				*									*		
A. implexa Benth.										*					
A. iteaphylla F.Muell. ex Benth.								*							
A. longifolia (Andrews) Willd.	*	*	*				*	*		*		*	*		
A. mangium Willd. #				*	*	*			*	*			*	*	
A. mearnsii De Wild. #		*	*	*	*	*	*	*	*	*	*	*	*	*	
A. melanoxylon R.Br. #	*	*		*		*	*	*		*	*	*	*		*
A. paradoxa DC.	*		*				*	*		*					
A. podalyriifolia A.Cunn. ex G.Don										*			*		
A. prominens A.Cunn. ex G.Don								*							
A. pycnantha Benth.		*						*		*					
A. retinodes Schltdl.		*				*									
A. salicina Lindl.			*									*			
A. saligna (Labill.) H.L.Wendl. #		*	*					*		*	*				
A. stricta (Andrews) Willd.										*					
A. verticillata A.Cunn.							*					*			
A. victoriae Benth.			*												

constraints of individual fitness levels) and helps populations overcome the negative effects of demographic and environmental stochasticity. For Australian acacias one thing is clear – all the wattles used extensively in forestry have become invasive somewhere in the world.

Of all the regions where wattles have been widely grown as alien species, South Africa provides a particularly useful case study for elucidating the role of multiple factors in shaping the biological and human dimensions of invasions. While these factors may be highly context-dependent, e.g. vegetation or climate type, many taxa do show similar trends under different contexts. Many wattle species were introduced to South Africa starting in the mid-1800s. *Acacia mearnsii*, *A. dealbata*, *A. decurrens* and *A. melanoxylon* were grown in plantations for timber, primarily for tannin production. Several species (notably *A. saligna*, *A. cyclops* and *A. longifolia*) were widely planted to stabilise sand dunes. A few other species were introduced and distributed as ornamental plants, e.g. *A. baileyana* and *A. podalyriifolia*. However, of the 80 or so species recorded as having been introduced into South Africa, most were only ever planted in small trial plots or arboreta, from which several have escaped and become invasive (Figure 1). Despite the observed problems of invasions caused by *A. dealbata*, *A. decurrens*, *A. mearnsii* and *A. melanoxylon*, these species have continued to be spread and planted around the world (Figure 2).

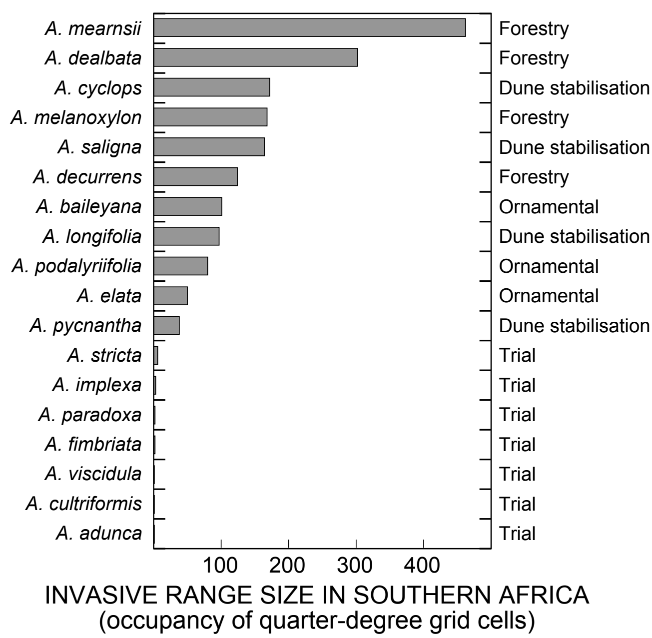

Figure 1: The extent of invasion by wattle species in South Africa is tightly linked to the reason the species were introduced and subsequently distributed by humans around the country. Data from the Southern African Plant Invaders Atlas (accessed May 2013)

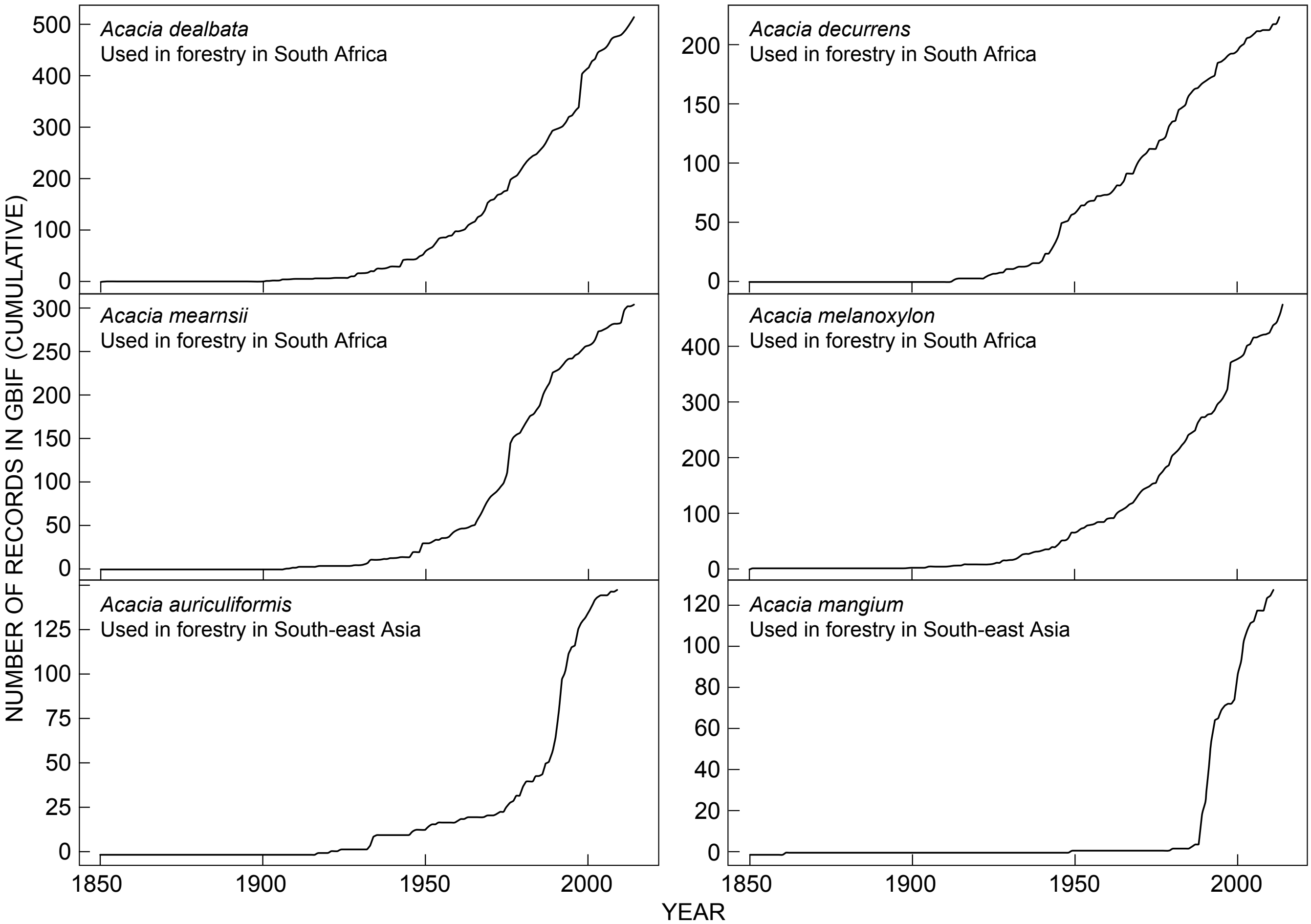

Figure 2: Australian *Acacia* species used for forestry in South Africa have had a much longer global history of introduction than species grown for forestry in South-east Asia. The plots show the cumulative number of records in the Global Biodiversity Information Facility (http://www.gbif.org) for the four main species historically used in South Africa forestry (*A. dealbata*, *A. decurrens*, *A. mearnsii* and *A. melanoxylon*) and the two main species used in forestry in South-east Asia (*A. auriculiformis* and *A. mangium*). The rise in use of *A. mangium* and *A. auriculiformis* in the last quarter of the twentieth century is clearly evident. Data were extracted using the function rgbif on 8 July 2014 (Chamberlain et al. 2014). Records from Australia, Papua New Guinea and Indonesia were omitted (as some of these species occur naturally in these countries), as well as records from Benin that skewed the observed pattern (over 3 000 samples were collected of *A. auriculiformis* from Benin during 2005–2007 presumably as part of a substantial molecular study)

A milestone in global wattle forestry was the shipment of more tropical species to South-east Asia (in particular, *A. auriculiformis*, *A. crassicarpa*, *A. mangium* and *A. mangium* × *A. auriculiformis* hybrid). This started in the early 1900s (for *A. auriculiformis*), increased rapidly between the mid- and late 1900s, and culminated in massive afforestation in the last couple of decades (Turnbull et al. 1998; Griffin et al 2011; Figure 2), although the drivers of forestry expansion differ between countries (Turnbull et al. 1998; Griffin et al. 2011; Kull et al. 2011). The environmental impacts of this huge new afforestation drive are a cause for concern for various reasons. The most obvious is the massive scale of habitat transformation with immediate implications for biodiversity. A less obvious concern relates to the potential invasiveness of these species. Although the issue of invasions associated with forestry is appreciated in many parts of the world (reviewed in Richardson et al. 2011), our perception after attending the conference on 'Sustaining the Future of *Acacia* Plantation Forestry' in Vietnam in March 2014 was that this issue enjoys very low priority in South-east Asia. There are several likely reasons for this. First, much afforestation in the region is on degraded land where concerns relating to possible environmental impacts due to invasions are insubstantial. As we understand it, the political priorities of the region hinge on rapid economic development rather than on sustainability or biodiversity conservation. Second, the plantings are relatively recent and so have not yet produced widespread invasions. Third, naturally recruiting individuals are currently utilised for fuel-wood, which may reduce the incidence of invasions. Fourth, differences between ecosystem level dynamics, in particular the predominance of fire as a driver of vegetation dynamics in South Africa, might mean that many areas of South-east Asia are inherently less invasible. Nonetheless, wattle invasions are likely to become more widespread in the

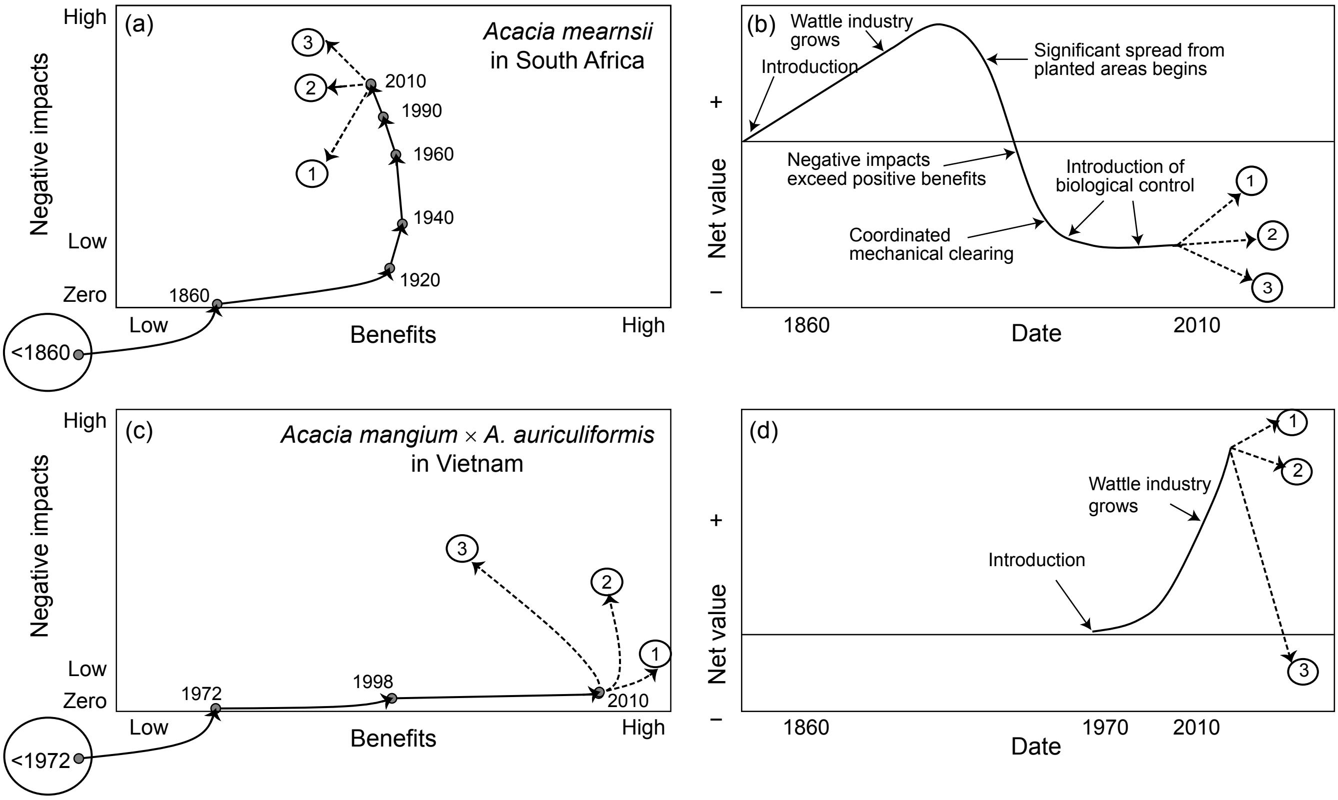

Figure 3: Historical and possible future costs and benefits associated with Australian *Acacia* introductions. (a, b) *Acacia mearnsii* was introduced to South Africa in 1860. The species initially provided substantial benefits through the production of timber and bark. However, costs began accruing when the species started spreading from plantations to invade natural and semi-natural vegetation where it has a range of impacts. There are various future scenarios: 1 = the optimum combination of management practices is fully implemented, and practices are effective; 2 = maintenance of the status quo, where the implementation of management practices is incomplete, not fully coordinated and sustained, or partially ineffective; 3 = the worst-case scenario, where key management practices are not implemented, or fail. (c, d) *Acacia mangium* × *A. auriculiformis* hybrids were introduced to Vietnam in the past 50 years, but the first substantial plantations only occurred at the end of the twentieth century (i.e. >100 years after plantations of *A. mearnsii* in South Africa). Since then there has been a massive increase in plantings, such that the relative contribution to the Vietnamese economy of these species is far in excess of that ever reached by Australian *Acacia* plantations in South Africa. No negative impacts have yet been recorded in Vietnam, but the future is uncertain: 1 = the forestry industry continues to be commercially viable, proactive efforts to prevent the build-up of large unmanageable seed-banks are implemented and spread to natural vegetation is limited; 2 = the negative effects of invasions start increasing and are only dealt with reactively, a substantial profitable forestry industry remains although it is no longer in a honey-moon period; 3 = there is a substantial reduction in the sustainability of Australian acacia forestry (due to changes in market demand and/or new pests and pathogens), plantings are left to spread unmanaged that result in large-scale undesirable ecosystem-level changes that are costly to reverse, the trees become a net cost to the region (as happened in South Africa). (a, b) Redrawn from van Wilgen et al. (2011) and van Wilgen and Richardson (2014); (c, d) based on data from Griffin et al. (2011), Harwood et al. (2015), Nambiar et al. (2015)

future, especially if the wattle forestry industry declines and current plantations are left unmanaged, or if people stop utilising recruits outside plantations.

The societal benefits and costs received from the new plantings in South-east Asia mirror the initial value obtained from plantings of a different suite of Australian acacias in South Africa well over a century ago (Figure 3). If the South African experience was to be repeated in Asia over the coming decades, not only would the invasions incur a substantial cost to society, but they would do so on a much larger scale. In our view, the sustainability of Australian acacia plantation forestry demands further elucidation of the drivers of invasions and that the management lessons learned from other regions need to be implemented proactively.

Drivers of invasiveness in wattles

Of the more than 80 species of Australian acacias introduced to South Africa, 18 have been recorded as naturalised (Table 1). The extent of invasion varies markedly between species and is closely linked to the reason for introduction and use of the species in South Africa – species used for forestry and dune stabilisation generally have much larger non-native ranges than those used for ornamentation or only used in trials or planted in arboreta (Figure 1).

The ultimate factors responsible for this pattern have been examined in various ways. As with invasive plants in South Africa in general, the extent of invasion (area occupied) for wattles is positively correlated with residence time (Wilson et al. 2007). However, in contrast to some other plant families, there is no phylogenetic signal for invasiveness of *Acacia* species, i.e. invasive taxa are dispersed throughout the phylogeny (Miller et al. 2011). Unlike the case with pines (Rejmánek and Richardson 1996), there is no clear evidence that any particular life-history syndrome separates the most invasive from less invasive species (Gibson et al. 2011). All species have traits that allow them to survive in small founder populations, disperse across natural and semi-natural ecosystems, and persist under a range of natural and human-induced disturbance regimes (Richardson and Kluge 2008; Gibson et al. 2011). To a large extent, human-use factors explain the extent of invasions of different species in the genus (Castro-Díez et al. 2011). Detailed studies of the trajectories of invasions of species with contrasting introduction and usage histories in South Africa also point to the driving role of human-orchestrated propagule pressure in mediating invasion dynamics across all major ecosystem types (Donaldson et al. 2014). Many other abiotic and biotic factors mediate the role of propagule pressure, notably fire, forest clearance and soil disturbance (Richardson et al. 1994). Taken together, the many studies of the invasion ecology of wattles point to a major 'invasion debt' throughout the introduced range of the group – many species have only begun to invade and are poised to invade much larger areas. This is the case even in South Africa where processes driving widespread invasions have played out for longer and over a greater extent than in other parts of the world.

Forestry practices can also lead indirectly to the selection of attributes that may enhance invasiveness. For example, a major focus in forestry research is genetic improvement through intentional approaches, such as hybridisation and polyploidisation, and unintentional processes, such as admixture (Le Roux et al. 2011; Harwood et al. 2015). Hybridisation has been linked to enhanced performance and subsequent invasiveness of species (Ellstrand and Schierenbeck 2000). In some instances, highly invasive lineages of wattles are the direct descendants of inter- and intra-specifically introgressed hybrids resulting from cultivation (e.g. *A. saligna*; Thompson et al. 2012) or already-admixed populations from Australia (e.g. *A. pycnantha*; Le Roux et al. 2013). Interactions between the environment and provenance are hugely important to foresters; multiple introductions from different parts of the native range of species are often included in field trials. Such co-introduction of previously allopatric genetic entities from the native range often leads to intra-specific hybridisation (admixture) in the introduced range, which may enhance performance and thus invasiveness (Le Roux et al. 2011; Zenni et al. 2014). Polyploidisation (chromosome numbers exceeding diploid numbers) is often desirable in forestry species because of the immediate effects on traits such as growth rate, wood density and pathogen resistance. However, it is clear that polyploidisation, whether within (autopolyploidisation) or between (allopolyploidisation) taxa across many plant families, is strongly linked to invasiveness as it confers immediate genetic advantages linked to genetic diversity and gene expression, physiological and environmental tolerance, altered biotic interactions, etc. (te Beest et al. 2012). In other words, fitness advantages associated with invasiveness as afforded by polyploidisation may not necessary entail higher fecundity (e.g. higher seed production or germination rates) but may often take the form of other life history traits advantageous under novel environmental conditions. For example, Griffin et al. (2012) found that tetraploid *Acacia mangium* had lower seed set compared to diploid genotypes but that tetraploid forms had higher levels of self-compatibility, a trait often associated with invasive plants (Rambuda and Johnson 2004). Many invasions are constrained by the absence of specialist mutualists, e.g. pollination and dispersal, in their new ranges (Traveset and Richardson 2014). For legumes, below-ground mutualist requirements, such as nitrogen-fixing symbiosis with rhizobia, may be particularly important with immediate effects on relative fitness (Rodríguez-Echeverría et al. 2011). For this reason effective co-evolved rhizobial inoculums are often developed for wattles in forestry plantations, alleviating mutualist constraints through the co-introduction of suitable rhizobia (e.g. *A. pycnantha* in South Africa; Ndlovu et al. 2013). Whether these co-introduced bacteria benefit invasiveness remains to be determined, but this seems likely.

Management lessons to be learned

South Africa has implemented the most comprehensive suite of interventions for dealing with invasive wattles (reviewed in van Wilgen et al. 2011). These interventions include measures to deal with both well-established invasive species with large ranges and species that have only recently begun to invade or have invaded only small areas. South Africa has also pioneered classical biological control of wattles, with agents – insects or fungi – introduced against most of the major invaders (Impson et al. 2009). These agents are playing a major role in reducing seed production and reducing the extent of invasions, with no non-target effects (most agents are restricted to a single Australian *Acacia* species or a narrow clade of Australian acacias; none have attacked native African acacias). Efforts to contain invasions of the major invasive species involve integrated control strategies, including mechanical, chemical and biological control practices as well as initiatives to achieve reductions in density and extent through utilisation (van Wilgen et al. 2011), and six species that have more limited distributions are currently targeted for eradication (Wilson et al. 2013), with substantial research conducted to determine the feasibility for eradication (e.g. Zenni et al. 2009; Moore et al. 2011). Despite a huge investment in the integrated control of woody invasive plants in South Africa (van Wilgen et al. 2012), and although success has been achieved in reducing the extent of invasions of some species in some areas, the overall extent of invasion of wattles is increasing. New legislation and the imminent publication of a far-reaching national strategy for biological invasions will require substantial changes to the way that invasive species are managed. One of the requirements will be national strategies for each major taxon of invasive species. To this end, van Wilgen et al. (2011) detailed a long list of elements required of such a national strategy to deal with the multiple dimensions of Australian acacia invasions. The huge extent of current invasions (e.g. Rouget et al. 2003), the complex conflicts of interest (Kull et al. 2011; van Wilgen and Richardson 2014) and other complexities associated with such national-scale management projects (e.g. Roura-Pascual et al. 2009) mean that problems with invasive wattles will remain a huge challenge for managers for decades to come.

In terms of policy response by the forestry industry, there are some general recommendations relating to invasiveness in some policies and regulatory frameworks around the world. For example, the Forestry Stewardship Council's (FSC) *FSC Guide to Integrated Pest, Disease and Weed Management in FSC Certified Forests and Plantations* (Willoughby et al. 2009) contains guidelines for reducing problems of invasive species. Most emphasis in such documents is, however, placed on reducing impacts of other invasive species on the forestry trees, rather than on invasiveness of the trees themselves. Indeed, a major shortcoming of the FSC scheme from a conservation perspective is that genetic modification is not allowed, despite the fact that induced sterility of forestry trees is a promising option for reducing invasiveness (Richardson and Petit 2005), an avenue that has been explored to some extent by traditional breeders (e.g. Griffin et al. 2015). Codes of conduct to reduce problems of invasions have been commissioned for horticulture and botanic gardens. For example, the Council of Europe and the Bern Convention have promoted several codes of conduct, including the European Code of Conduct for Botanic Gardens on Invasive Alien Species, the Code of Conduct on Horticulture and Invasive Alien Plants, and the European Code of Conduct on Pets and Invasive Alien Species. Further attention is required to ensure that enterprises relying on inherently invasive species (such as forestry using wattles) must undertake to implement measures to reduce or eliminate problems associated with invasiveness.

In summarising lessons to be learnt from efforts at controlling invasive wattles in different parts of the world, Wilson et al. (2011) suggested priorities for the proactive management of Australian acacias to prevent invasions. Of those listed by Wilson et al. (2011) we consider the following to be particularly relevant in the Asian context: '(1) All new introductions should be contingent upon full and detailed risk assessments and cost-benefit analyses' (while perceived benefits might well override ecological concerns in some cases, risk quantification could trigger the development of sustainable mitigation methods); (2) 'Commercial plantings should carry the costs for the increased risk of invasions'; (3) 'Production should focus on sterile cultivars, and responsible utilization and containment practices should be developed and implemented' (e.g. see Griffin et al. 2015); (4) Biological control is a cost-effective, sustainable and reliable option for invasive wattles (as has been shown from experiences in South Africa); and (5) Effective management of wattle invasions requires sharing of information and experience, and an increase in public awareness.

An immediate priority is to assess the current extent of invasions to provide a baseline for monitoring. A spatially-explicit risk analysis (along the lines of Rouget et al. 2002) could identify zones to be prioritised in monitoring. We recommend the establishment of a network of sentinel sites for monitoring; Visser et al. (2014) show that Google Earth images are useful for monitoring tree invasions in some cases.

Conclusions

While substantial advances have been made towards understanding biological invasions, much uncertainty exists about which species will become invasive and which will have major impacts. A robust generalisation that can be made is that experiences in one part of the world are useful for predicting outcomes in other regions. The recent wave of plantings of Australian acacias in South-east Asia has already produced invasions, but the extent of these invasions and associated impacts is set to increase substantially. The negative experiences of wattle invasions in South Africa, and the subsequent negative publicity surrounding the wattle industry, need not be repeated in South-east Asia. The main challenge is whether there will be political buy-in to address the problem proactively.

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